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COMPARISON OF MACROINVERTEBRATE-DERIVED STREAM QUALITY METRICS BETWEEN SNAG AND RIFFLE HABITATS[1]

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ABSTRACT: We compared benthic macroinvertebrate assemblage structure at snag and riffle habitats in 43 Wisconsin streams across a range of watershed urbanization using a variety of stream quality metrics. Discriminant analysis indicated that dominant taxa at riffles and snags differed; Hydropsychid caddisflies (*Hydropsyche betteni* and Cheumatopsyche spp.) and elmid beetles (*Optioservus* spp. and *Stenemlis* spp.) typified riffles, whereas isopods (*Asellus intermedius*) and amphipods (*Hyalella azteca* and *Gammarus pseudolimnaeus*) predominated in snags. Analysis of covariance indicated that samples from snag and riffle habitats differed significantly in their response to the urbanization gradient for the Hilsenhoff biotic index (BI), Shannon's diversity index, and percent of filterers, shredders, and pollution intolerant Ephemeroptera, Plecoptera, and Trichoptera (EPT) at each stream site ($p \leq 0.10$). These differences suggest that although macroinvertebrate assemblages present in either habitat type are sensitive to detecting the effects of urbanization, metrics derived from different habitats should not be intermixed when assessing stream quality through biomonitoring. This can be a limitation to resource managers who wish to compare water quality among streams where the same habitat type is not available at all stream locations, or where a specific habitat type (i.e., a riffle) is required to determine a metric value (i.e., BI). To account for differences in stream quality at sites lacking riffle habitat, snag-derived metric values can be adjusted based on those obtained from riffles that have been exposed to the same level of urbanization. Comparison of nonlinear regression equations that related stream quality metric values from the two habitat types to percent watershed urbanization indicated that snag habitats had on average 30.2 fewer percent EPT individuals, a lower diversity index value than riffles, and a BI value of 0.29 greater than riffles.

(KEY TERMS: land use; macroinvertebrate community assemblages; streams; riffles; snags; urbanization; discriminant analysis; imperviousness.)

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INTRODUCTION

Anthropogenic disturbances in a watershed, particularly those associated with urban development, are known to affect macroinvertebrate community assemblages. Shifts in structure and function, including changes in composition, richness, abundance, functional feeding groups, and diversity have been reported from streams subject to such disturbances (Pitt and Bozeman, 1983; Pedersen and Perkins, 1986; Jones and Clark, 1987; Richards and Host, 1994; Lenat and Crawford, 1994; Elliott *et al.*, 1997). As such, benthic macroinvertebrates have been shown to be useful in assessing the health of aquatic systems (Kaesler and Cairns, 1972). Not only are these organisms ubiquitous, not very mobile, and abundant, making them ideal bioindicators, but most or a portion of their life cycle is aquatic and they can be found in a wide variety of habitat types. These properties offer the opportunity to assess temporal changes in stream quality rather than a snapshot estimate which is typically provided by chemical or physical assessments (Rosenberg and Resh, 1993).

Numerous metrics have been developed to describe stream quality based on abundance and composition of macroinvertebrate populations (Shannon, 1948; Margalef, 1958; Chutter, 1972; Hilsenhoff, 1977; Chesters, 1980; Plafkin *et al.*, 1989; Rosenberg and Resh, 1993). However, a variety of site-specific attributes may affect these stream quality metrics because of their influence on macroinvertebrate communities. These attributes include substrate size and type (Hynes, 1970a,b; Allan, 1975; Minshall and Minshall, 1977; Reice, 1980; Williams, 1980; Erman and Erman, 1984; Clements, 1987), velocity (Minshall and Minshall, 1977; Williams, 1980; Erman and Erman, 1984), habitat type (Minshall and Minshall, 1977), and physical and chemical properties (Baker and Sharp, 1998).

Understanding how site-specific habitat attributes affect macroinvertebrate assemblages and their calculated stream quality metrics is crucial in biomonitoring. This is particularly important where habitat differences cannot be standardized because of inherent differences in geomorphic or watershed characteristics or where the type of available habitat is limited or altered as a result of development or disturbance in a watershed (Malmqvist and Rundle, 2002). A better understanding of these differences may enable adjustment of the metric values across habitat types to maximize their potential use. This would benefit resource managers in their decision making processes by allowing them to compare water quality among streams where the same habitat type is not available at all stream locations, or where a specific habitat type (i.e., a riffle) is required to determine a metric value [i.e., biotic index (BI)].

Stream quality assessments using macroinvertebrates often rely on metrics based on data from standardized samples collected from riffles (Plafkin *et al.*, 1989). However, riffles are not always present, thus investigators often substitute other habitat types such as snags (defined herein as branches or other debris onto which organic material has collected). Although differences in macroinvertebrate composition between riffle and pool habitats or among different rocky substrate sizes have been documented (e.g., Minshall and Minshall, 1977; King, 1981; McCulloch, 1986; Quinn and Hickey, 1990; Brown and Brussock, 1991; Payne and Miller, 1991), only limited studies have evaluated differences in macroinvertebrate collections between riffle and snag habitats (Hooper, 1993; Wang *et al.*, 2006). More studies are especially needed to evaluate the influences of such different habitats on the macroinvertebrate metrics and how to adjust for these differences in water quality assessments. To address this, our first objective was to confirm differences in macroinvertebrate taxonomic assemblage between habitat types at our study sites. Our second objective was to assess differences in stream quality metrics between snag and riffle habitats in these southeastern Wisconsin streams subject to a gradient of watershed urbanization. These were a subset of those streams studied by Wang *et al.* (2006). Our third objective was to determine adjusted snag metric scores that could be used to evaluate stream quality in systems without riffles as compared to those with riffles that were subject to the same level of urbanization (or vice versa). These adjusted scores can be valuable to resource managers who wish to make comparisons of stream quality assessments between sites with different available habitat types.

Study Area

We assessed watershed land use and macroinvertebrate assemblages at 43 stream sites in Southeastern Wisconsin (Figure 1). All streams were located in unique subwatersheds in the Southeastern Wisconsin Till Plains Ecoregion (Omernik and Gallant, 1989). Most of this area has low relief and slopes are predominantly level to slightly rolling. The stream drainage system is poorly developed and undrained depressions are common. Wang *et al.* (2000) provided a detailed description of each study stream. Because of nutrient rich soils and low relief, this ecoregion was historically an important agricultural area, but by 1990 about two million people lived in this region, and agricultural land use had decreased.

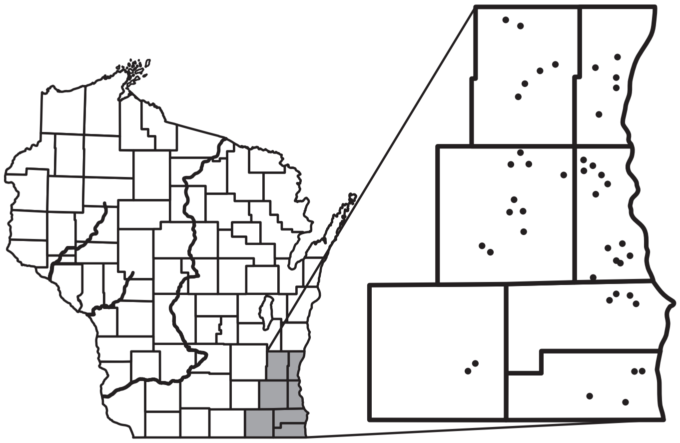

FIGURE 1. Location of Forty-Three Stream Sites in Southeastern Wisconsin.

Three criteria were used to select the study sites. First, to minimize natural variation in biotic communities, sites with similar watershed soil types, stream size and slope, and natural hydrological and temperature regimes were selected. They represented a range of watershed urban development (as measured by imperviousness) and environmental quality, from least impacted regional reference conditions to heavily degraded conditions (Figure 2). Second, sites were chosen in watersheds for which detailed land use data could be obtained. Finally, sites with historical fish population data were selected to meet requirements of a concurrent study about the effects of urbanization on fish communities (Wang *et al.*, 2000, 2001).

Subwatershed areas ranged between 5.55 and 101.73 km^2, with a mean of 28.33 km^2. Impervious surfaces within subwatersheds (a measure of perturbation in urban watersheds; Schueler, 1994) ranged from 1.9 to 46.3% with a mean of 12.0%. To minimize variation in potential natural biological attributes, all sampling sites were warmwater (summer maximum daily mean temperature exceeded 24°C), second to third order in size, and had low to moderate gradients (< 6 m/km). These stream sites, in the absence of differing watershed land uses, were expected to have similar habitat and biological communities (Lyons, 1996).

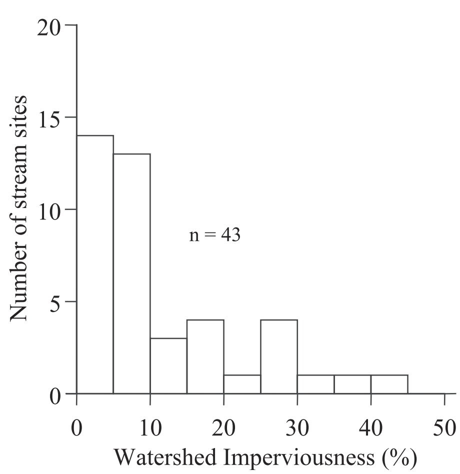

FIGURE 2. Distribution of Sampling Sites Across a Gradient of Percent Watershed Imperviousness.

METHODS

Macroinvertebrates

In October 1997, we sampled aquatic macroinvertebrates from riffle and snag habitats separately using a 600 μm mesh D-frame kick net. Riffles were defined as areas where water velocity was greatest (preferably 0.3 m/s or greater) and the surface was turbulent because of a decrease in depth. Bottom substrates were composed of cobble, gravel, small rocks, or other stable substrate of fairly uniform size. Snags were defined as partially or fully submerged accumulations of vegetative debris (e.g., leaves, grasses) on branches, logs or other objects within the stream channel.

During sampling, we approached each habitat type from downstream to minimize disturbance prior to positioning the D-frame net. At riffles, we positioned the D-frame net on the stream bottom and vigorously disturbed the substrate just upstream from the net by kicking. This process dislodged macroinvertebrates, allowing them to be washed into the net (Hilsenhoff, 1987). At snags, we positioned the D-frame net in the water column where it would collect most of the dislodged debris. Depending on water depth, we disturbed the snag using either hands or feet. If the net became clogged, we cleared it and continued the collection process.

At each site, we repeated this process at three locations for each habitat type. We sampled riffles across a transect perpendicular to flow. If the stream was too narrow for three successive samples to be collected, we collected them by moving to an upstream riffle. We collected snag samples in areas of highest water velocity with first consideration given to largest snags. We sampled additional snags with similar consideration given to size of snag and water velocity.

For each habitat type, we combined the three samples to ensure that at least 125 organisms with assigned BI tolerance values were obtained

(Hilsenhoff, 1987). This number of organisms is usually present when enough debris to fill an 8-ounce jar has been collected (Hilsenhoff, 1987). We visually inspected debris in the field to assess general number of organisms collected. At highly degraded sites where macroinvertebrates or appropriate sampling habitat types were rare, a more extensive sampling effort was required. In such cases, regardless of the number of macroinvertebrates collected, we stopped sampling after one hour or after a standard habitat sampling reach (35 times mean stream width; Lyons, 1992) had been assessed.

By hand or using forceps, we removed organisms found clinging to the net or to large organic debris and placed them in sample collection jars. We then discarded the large organic debris at the field site. We preserved macroinvertebrate samples in 80% isopropyl alcohol, and transferred the samples to fresh alcohol after 24 and 48 hours. We later transported samples to the laboratory for sorting and identification.

In the laboratory, we placed macroinvertebrates and debris in a Pyrex® plate positioned over a grid of twelve 6.5 cm^2 squares. Using a random number table, we chose a grid square from which to remove invertebrates. With the aid of an illuminated magnifying lamp, we removed all invertebrates from the grid square. Organisms that had BI tolerance values were enumerated. We repeated this process in additional, consecutively numbered grid squares until a minimum of 125 organisms with BI tolerance values had been enumerated or until the entire sample had been sorted.

We identified macroinvertebrate taxa to the lowest taxonomic level for which a BI tolerance value was defined (Hilsenhoff, 1987). We identified organisms without defined BI tolerance values to the lowest level of taxonomic resolution possible. We slide-mounted chironomid larvae in CMC-10® mounting media (Master's Chemical Company, Inc., Bensenville, IL) to increase clarity of specimens, and allowed them to clear for two days prior to identifying them.

Watershed Land Use

We used percent watershed imperviousness as a surrogate for degree of urbanization using a Geographic Information System database (developed by the Southeastern Wisconsin Regional Planning Commission, 1990). The database was developed from 1:4,800 air photos. Nine major land use categories were defined: residential, commercial, industrial, transportation, communication and utilities, government and institutional, recreational, agricultural, and open lands. Each of these categories included sub-categories, for a total of 63 possible land use types. Each land use type was assigned a percent effective connected imperviousness based on surfaces such as roads, rooftops, and parking lots. These surfaces were typically connected to surface waterways by some direct route, such as a storm sewer, drainage pipe, or surface drainage way (Booth and Jackson, 1997). Watershed imperviousness was calculated for each land use type, summed and standardized by watershed area. Watershed boundaries and upstream land area were determined following methods described in Wang *et al.* (2000).

Analyses

For our assessments, we calculated several stream quality metrics (Plafkin *et al.*, 1989) based on macroinvertebrate data, including Hilsenhoff's BI (Hilsenhoff, 1987), the Shannon's diversity index (Shannon, 1948), generic and species richness (Gaufin, 1957), and percent pollution intolerant Ephemeroptera, Plecoptera, and Trichoptera (EPT). We calculated percent collectors, filterers, gatherers, scrapers, and shredders to describe functional feeding response to increasing levels of watershed imperviousness (Merritt and Cummins, 1996). Although we recognize that there is a difference between functional feeding groups, diversity indices, and other stream quality assessment methods, for purposes of simplicity and following Plafkin *et al.* (1989), we refer to these collectively as stream quality metrics in this paper. Metrics calculated for both habitat types were plotted vs. percent watershed imperviousness to assess the impact of urbanization on the macroinvertebrate communities (Stepenuck *et al.*, 2002) and to compare differences in such impacts between habitat types.

To develop snag-adjusted metric values to assess stream health at sites without riffles, we first confirmed that macroinvertebrate community assemblages differed between the sampled habitats. This was necessary because values of some metrics were dependent upon the taxa that were identified in a sample (i.e., Hilsenhoff BI). To confirm that macroinvertebrate assemblages in snags could be discriminated from those at riffles, we analyzed data using canonical discriminant function analysis (SAS Institute Inc., 1992) in an approach similar to Tonn *et al.* (1983). Final classification success rates were obtained using a split-half validation process with priors set proportional. We only conducted analyses on taxa comprising at least 2% of the entire assemblage to reduce influence of rare species. At sites where two snag samples were collected (i.e., where riffles were not present) we averaged metric values to be used in the discriminant function analysis.

We then assessed metric values to evaluate if the slopes of the regressions of the observed relationships (i.e., between a metric value and percent watershed imperviousness) were uniform between habitat types. When slopes were not uniform, or a significant interaction ($p \leq 0.10$) existed between the covariate (i.e., imperviousness) and the treatment (i.e., habitat type), we did not perform Analysis of Covariance (ANCOVA). In such cases, we could not determine a snag-adjusted value.

For metrics in which no significant interaction existed between the covariate and the treatment, we performed subsequent ANCOVA analyses (SAS Institute Inc., 1992; SPSS Inc., 1998). This allowed us to determine significant differences between snag and riffle habitats in stream quality metrics along a gradient of watershed imperviousness.

Prior to ANCOVA, we transformed metrics to linearize the relationship with the covariate. $\text{Log}_e(x)$ was used for most metrics, however, we utilized $\log_e(x + 1)$ to correct for zeroes in the data sets for percent EPT, and percent scrapers, filterers, and shredders. We considered the relationship between a metric and percent watershed imperviousness significant at $p \leq 0.10$.

When ANCOVA revealed significant differences between treatments as they covaried with watershed imperviousness, we used the differences between the intercepts of the nonlinear regression equations as snag-adjusted values between the treatments. We reported these snag-adjusted values as the difference in value or percent for each of the metrics.

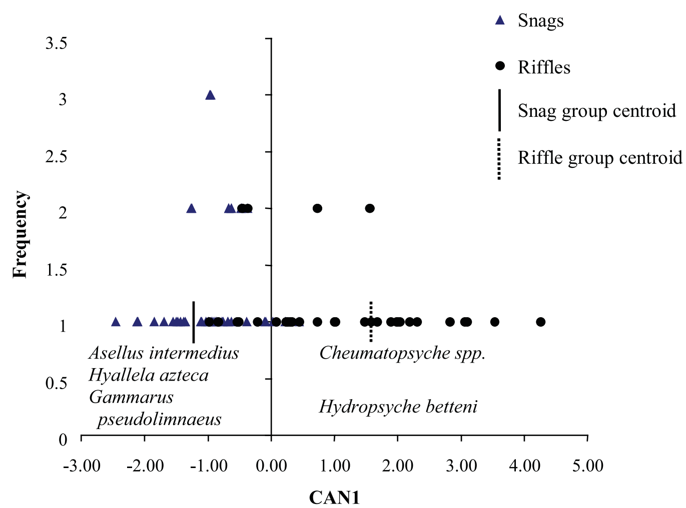

FIGURE 3. Canonical Discriminant Axis Discriminating Macroinvertebrate Species at Riffle vs. Snag Habitats.

TABLE 1. Structure Coefficients for the Canonical Discriminant Function.

Taxa	CAN 1
Cheumatopsyche spp.	0.607
Hydropsyche betteni	0.634
Optioservus spp.	0.493
Stenelmis spp.	0.582
Simulium vittatum	0.327
Gammarus pseudolimnaeus	−0.240
Hyalella azteca	−0.398
Asellus intermedius	−0.289
Turbellaria spp.	−0.127
Oligochaeta spp.	0.274
Physa spp.	−0.057

RESULTS

Discrimination of taxonomic assemblages between snag and riffle habitats was significant (Wilks' $\lambda = 0.4857$, $p < 0.0001$) and resulted in an initial correct classification rate of 85.9% (Figure 3). Nearly unbiased split-half validation of the data resulted in an 80.8% correct classification rate in assignment of habitat type using the discriminant functions defined in the initial classification. Hydropsychid caddisflies [*Hydropsyche betteni* (Ross) and *Cheumatopsyche* spp. (Wellengren)] and elmid beetles [*Optioservus* spp. (Sanderson) and *Stenemlis* spp. (Dufour)] characterized riffle habitat with positively loaded structure coefficients in the discriminant function. Isopods [*Asellus intermedius* (Forbes)] and amphipods [*Hyalella azteca* (Saussure) and *Gammarus pseudolimnaeus* (Bousfield)] represented snag habitat with high negative loadings (Table 1).

ANCOVA revealed that many macroinvertebrate metrics were significantly ($p \leq 0.10$) related to the degree of imperviousness in the watershed. For example, differences existed between treatments (i.e., snag vs. riffle habitats) as they covaried with the watershed imperviousness for BI, Shannon's diversity index, and percent EPT, shredders, and filterers (Table 2). Species and generic richness were not significantly related to degree of imperviousness in the watershed. The BI score was consistently lower in riffles than in snags by an average of 0.29, indicating better inferred stream quality in riffles across the range of watershed imperviousness (Figure 4, Table 3). Shannon's diversity index was consistently higher in riffles than in snags by a value of 0.58 (Figure 4, Table 3). Riffles had an average of 30.2% more EPT individuals (Figure 4, Table 3) and 44.2% more filterers (Figure 5, Table 3) than snags in our study watersheds. There were 1.5% more shredders in snags across the imperviousness gradient (Figure 5, Table 3). Slopes of the regressions were not uniform

TABLE 2. Nonlinear Regression Parameter Estimates From Analysis of Covariance Results.

Parameter	Coefficient	SE	t	Probability
BI	(r^2 = 0.315 n = 78)			
Intercept	1.600	0.059	27.04	0.000
Habitat type	−0.105	0.042	−2.52	0.014
Diversity	(r^2 = 0.204 n = 78)			
Intercept	1.121	0.131	8.580	0.000
Habitat type	0.224	0.092	2.428	0.018
EPT individuals (%)	(r^2 = 0.295 n = 78)			
Intercept	3.211	0.390	8.232	0.000
Habitat type	1.059	0.276	3.838	0.000
EPT genera (#)	(r^2 = 0.413 n = 78)			
Intercept	2.166	0.162	13.378	0.000
Habitat type	0.239	0.115	2.085	0.040
Species richness	(r^2 = 0.120 n = 78)			
Intercept	3.079	0.124	24.771	0.000
Habitat type	−0.117	0.088	−1.328	0.188
Generic richness	(r^2 = 0.110 n = 78)			
Intercept	3.027	0.123	24.570	0.000
Habitat type	−0.120	0.087	−1.380	0.172
Filterers (%)	(R^2 = 0.337 N = 78)			
Intercept	2.394	0.359	6.670	0.000
Habitat type	1.476	0.256	5.771	0.000
Shredders (%)	(R^2 = 0.147 N = 78)			
Intercept	1.473	0.225	6.536	0.000
Habitat type	−0.368	0.161	−2.290	0.025

Note: Nonlinear regression parameter estimates are for $\log_e$ (stream quality metrics) vs. $\log_e$ (% watershed imperviousness) between snag and riffle habitats.

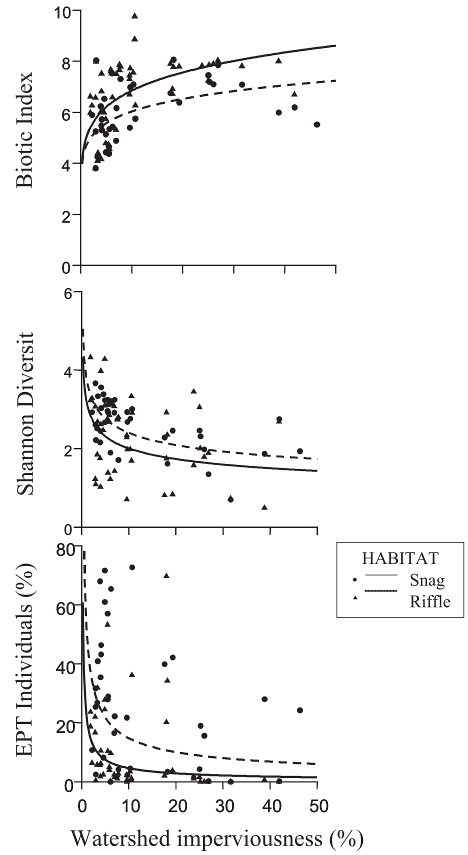

FIGURE 4. Fitted Nonlinear Regression Models Relating BI, Shannon's Diversity Index, and Percent EPT to Percent Watershed Imperviousness for Riffle and Snag Habitats.

for percent collectors, gatherers, and scrapers, thus a snag-adjusted value could not be determined.

DISCUSSION

Numerous macroinvertebrate metrics clearly were affected by urbanization that was represented by watershed imperviousness in this study. Watershed urbanization negatively affected BI, Shannon's diversity index, percent EPT, and generic richness. These results concur with the findings of Klein (1979), Lenat and Crawford (1994), Shaver *et al.* (1995), Horner *et al.* (1996), and May *et al.* (1997) and are discussed in detail in Stepenuck *et al.* (2002).

Significant differences between macroinvertebrate communities inhabiting snag and riffle habitats existed in stream quality assessments in our southeastern Wisconsin study streams. These differences were similar to findings by Humphries *et al.* (1998) and Parsons and Norris (1996) who also found differences in macroinvertebrate community assemblages between habitats. Stream quality metrics calculated from samples collected from snag habitats consistently indicated more degraded stream quality than those calculated from samples collected at riffle habitats at the same site. It is likely, therefore, that the physical habitat in these streams directly affects the estimation of water quality attributes based on the macroinvertebrates. Because differences in community assemblage between habitat types at the same stream site may lead to varied stream quality assessments when water quality or stream health is evaluated, metrics derived from different habitats on the same or different streams should not be intermixed or compared without adjusting the scores. Wang *et al.* (2006) suggested using snags as an alternate sampling substrate for streams without riffles. However, this approach does not provide managers with the ability to compare water quality scores between high and low gradient streams. To address this issue, we suggest that snag-adjusted values for metrics shown to differ between snag and riffle habitats may be utilized by managers to evaluate stream health in watersheds without riffles or in which snags were sampled. Using snag-adjusted stream quality metrics could be a valuable management tool in ecoregions that are defined by low gradient, channelized streams, such as those commonly found in

TABLE 3. Nonlinear Regression Equations Relating Snag and Riffle Habitats to Percent Watershed Imperviousness.

Nonlinear Regression Equation	r^2	p	Habitat Type
BI = 4.94 imperviousness (%) $^{0.142}$	0.28	0.0001	Snag
Shannon's diversity = 3.24 imperviousness (%)$^{-0.208}$	0.11	0.0172	Snag
Percent EPT = 23.0 imperviousness (%)$^{-0.701}$	0.21	0.0008	Snag
Percent filterers = 9.91 imperviousness (%)$^{-0.402}$	0.07	0.061	Snag
Percent shredders = 4.44 imperviousness (%)$^{-0.287}$	0.07	0.058	Snag
BI = 4.65 imperviousness (%) $^{0.113}$	0.25	0.002	Riffle
Shannon's diversity = 3.82 imperviousness (%)$^{-0.201}$	0.31	0.0004	Riffle
Percent EPT = 53.20 imperviousness (%)$^{-0.557}$	0.14	0.023	Riffle
Percent filterers = 54.11 imperviousness (%)$^{-0.507}$	0.14	0.026	Riffle
Percent shredders = 2.96 imperviousness (%)$^{-0.269}$	0.16	0.017	Riffle

Note: Nonlinear regression equations relating snag and riffle habitats to % watershed imperviousness are for BI, Shannon's diversity, % EPT, % filterers, and % shredders.

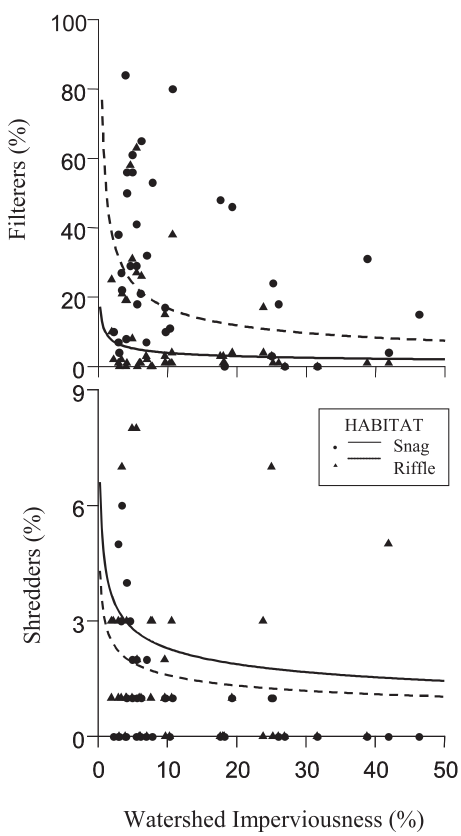

FIGURE 5. Fitted Nonlinear Regression Models Relating Percent Filterers and Shredders to Percent Watershed for Riffle and Snag Habitats.

the upper Midwestern United States (Omernik and Gallant, 1989).

Resources managers in Wisconsin have reported difficulty locating riffles from which to sample both because of hydrologic modification and because of naturally low gradient streams (S. Galarneau and M. Miller, Wisconsin Department of Natural Resources, February 2007, personal communication). In fact, of 9,650 macroinvertebrate samples collected by Wisconsin Department of Natural Resources staff between 1975 and 2006, 21% were collected from snags, runs, or pools (64% were collected from riffles, which is the prescribed habitat from which to sample whenever possible) (UWSP Aquatic Entomology Lab, 2007). Thus, snag-adjusted metrics could provide managers valuable data to allow for comparisons between sites for a significant portion of their sampling efforts. With limited funding to implement water quality improvement projects, such comparison between sites could help managers prioritize locations at which to implement these projects.

We determined snag-adjusted metric values for BI, Shannon's diversity index, and percent EPT, shredders and filterers. Supplementary studies need to be done, however, to determine if the adjusted values identified in this study are valid for other watersheds along a similar gradient of percent watershed imperviousness. Because of the diversity of habitats present in stream ecosystems, it would be highly useful to derive other nonlinear regression equations for macroinvertebrate metric scores from samples collected from habitats other than snags.

Other research has not addressed adjusting macroinvertebrate community metric scores from snags (or other habitats) to riffles based on regression coefficients across a gradient of watershed imperviousness. Some techniques used to account for differences in habitat types have included sampling at all available habitats and compositing samples to a single sample (Wright *et al.*, 1984), and developing unique sampling methods for high and low gradient streams (Kentucky Division for Environmental Protection, 2002). These techniques tend to be more time intensive, and do not necessarily allow for direct comparison of stream quality across habitat types. However, with increased use and resolution of Geographic Information Systems and the accepted relation between watershed imperviousness and water quality (Schueler, 1994; Arnold and Gibbons, 1996), refined measures of imperviousness will enhance our ability to develop better nonlinear regression models within different ecoregions for future stream quality assessment studies.

Differences in stream quality between snag and riffle habitats were apparent, however, direct comparison of these differences using nonlinear regression models allowed us to calculate adjusted values for metrics that are more useful when specific habitat types are not necessarily present in a stream. Successful implementation of our regression model can help to simplify some of the common problems associated with calculating macroinvertebrate stream quality metrics from different habitat types. The approach used will allow metric values to be adjusted for samples collected at the most available habitat types, rather than limiting sample collection to sites with riffles.

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